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Integrating biodiversity, ecosystem services and socio-economic data to identify priority areas and landowners for conservation actions at the national scale

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ABSTRACT

Gaps in research exist for country-wide analyses to identify areas of particular importance for biodiversity and ecosystem services to help reach Aichi Target 11 in developing countries. Here we provide a spatial conservation prioritization approach that ranks landowners for maximizing the representation of biodiversity features and ecosystem services, while exploring the trade-offs with agricultural and commercial forestry production and land cost, using Uruguay as a case study. Specifically, we explored four policy scenarios, ranging from a business as usual scenario where only biodiversity and ecosystem services were included in the analysis to a potentially unsustainable scenario where expansion of alternative land uses and economic development would be given higher priority over biodiversity and ecosystem services. At the 17% land target proposed for conservation, the representation levels for biodiversity and ecosystem services were, on average, higher under the business as usual scenario. However, a small addition to the proposed target (from 17 to 20%) allowed to meet same representation levels for biodiversity and ecosystem services, while decreasing conflict with agricultural and commercial forestry production and opportunity costs to local landowners. Under the unsustainable scenario, a striking 41% addition to the conservation target (from 17 to 58%) was needed to meet same representation levels for threatened ecosystems and ecosystem services, which are crucial to sustain human well-being. Our results highlight that more realistic and potentially higher conservation targets, than politically set targets, can be achieved at the country level when sustainable development needs are also accounted for.

1. Introduction

Current declines of biodiversity and ecosystem services are unprecedented (Butchart et al., 2010). Local, national and international policies have been promoted and are being implemented to halt and reverse such declines. In 2010, 20 Aichi targets were adopted by the Convention of Biological Diversity to address this challenge (Convention on Biological Diversity, 2010). Aichi Target 11 promotes the expansion of the global protected area network to cover 17% of all terrestrial land by 2020. Individual countries have committed to conserve 3–50% of their land to help reach this target (Butchart et al., 2015). Decision makers need effective methods and scientifically sound information to identify "areas of particular importance for biodiversity and ecosystem services" through "ecologically representative systems of protected areas and other effective area-based conservation measures" (Convention on Biological Diversity, 2010). The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) promotes the use of scenarios to assess various policy interventions in order to inform Aichi Target 11 (IPBES, 2016). Meanwhile, the UN Sustainable Development Goal 15 (Life on land), which directly links to Aichi Target 11, promotes the integration of ecosystem and biodiversity values into national and local planning, development processes, and poverty reduction strategies (http://www.un.org/sustainabledevelopment/biodiversity/).

Spatial conservation prioritization is the sub-field of conservation planning that deals with the identification of priority areas for conservation action where limited resources should be allocated (Moilanen et al.,

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2009). Spatial conservation prioritization can be carried out at multiple scales, ranging from global to local (Butchart et al., 2015; Di Minin et al., 2016, 2013; Game et al., 2011; Montesino Pouzols et al., 2014; Smith et al., 2016; Soutullo et al., 2008; Venter et al., 2014). A recent global analysis found that expanding the global protected area network to 17% of all terrestrial land could potentially triple the coverage of all terrestrial vertebrate species if countries were to collaborate in the identification of new protected areas, as opposed to acting independently (Montesino Pouzols et al., 2014). At the same time, national analyses remain crucial, as countries are identified as the main actors in the implementation of the Aichi Targets (Convention on Biological Diversity, 2010). Furthermore, national to regional conservation planning assessments can include data that may not be readily available at continental extents, including detailed information about social, economic, and political factors affecting on-the-ground implementation (Knight et al., 2006). National to regional conservation planning assessments can also help identify priority areas that can fulfil and sustain the local demand for ecosystem services by their human beneficiaries (Cimon-Morin et al., 2013).

Conservation planning assessments published in scientific literature are mainly from Europe, North America, Oceania and South Africa (Kukkala and Moilanen, 2013; Kullberg and Moilanen, 2014). Currently, gaps in research exist for country-wide analyses at a fine resolution that encompass the full set of biological and socio-economic data needed to inform conservation decision-making in developing countries (Kukkala and Moilanen, 2013; Kullberg and Moilanen, 2014). This is worrying, as developing countries host some of the most threatened biodiversity (Butchart et al., 2015; Montesino Pouzols et al., 2014) and ecosystem services (Turner et al., 2007). Nation-wide conservation planning assessments are mostly missing from South America where there has mainly been a focus on regional scale conservation planning within countries (see e.g. Faleiro and Loyola, 2013; Faleiro et al., 2013; Tognelli et al., 2008). The lack of comprehensive, high-resolution, up-to-date spatial information about species, ecosystems, and ecosystem services is a major constraint to the development of conservation planning assessments in developing countries (Di Minin and Toivonen, 2015; Stephenson et al., 2016). In addition, conservation planning assessments often ignore the needs of society for human and economic development and food production (Lambin and Meyfroidt, 2011), failing to be implemented (Knight et al., 2006). Finally, national conservation planning assessments should be more directly linked to land use planning in order to promote stakeholders' engagement and enhance the implementation of conservation actions (Pierce et al., 2005).

To our knowledge, no previous country-wide conservation planning assessment to identify priority areas for the conservation of biodiversity and ecosystem services to help reach Aichi Target 11 was developed for a South American country. Here, we fill this gap and provide a spatial conservation prioritization approach for maximizing the representation of biodiversity features and ecosystem services, while exploring the trade-offs with agricultural and commercial forestry production and land cost, using Uruguay as a case study. Importantly, our approach can be used to identify the most important landowners to engage in the implementation of conservation actions at the national level. In Uruguay, national conservation authorities have independently identified key biodiversity features and ecosystem services that the country needs to conserve, and generated updated information on the spatial distributions of the same, with the aim of identifying priority areas for the expansion of Uruguay's presently very limited protected area network (< 1% of Uruguay is protected) to help reach Aichi target 11 (MVOTMA, 2015). The conservation authorities also aim to include the strategy for protected area expansion within a broader strategy for sustainable development following the UN Sustainable Development Goals, particularly #15 of which Aichi Target 11 is part of. In order to do so, we explored four policy scenarios to assess whether it was possible to decrease conflict between conservation and alternative land uses, as well as opportunity costs to local landowners.

2. Methods

2.1. Study area

Located in temperate South America (Fig. 1), most of Uruguay is a rolling plain that represents a transition from the Argentinian pampas to the hilly uplands of southern Brazil. Uruguay has a humid subtropical climate that is fairly uniform nationwide (MVOTMA, 2010). The whole country is part of the Uruguayan Savannah ecoregion, which is classified as a ‘crisis’ ecoregion because of its extensive habitat conversion and limited habitat protection (Hoekstra et al., 2005). This ecoregion constitutes one of the richest areas for grassland biodiversity in the world, including vegetation communities of great species diversity (~2000 species). Temperate grasslands, which are the most threatened biome at the global level with <0.6% of its extent protected (Noss, 2013), are the main ecosystem in Uruguay.

Historically, traditional cattle-ranching on native grasslands has been the main economic activity. Over the last 15 years a sustained increase in commercial forestry and agriculture has challenged meat as the main export product. With a population of <3.5 million people with <20 people per km^2 (http://www.ine.gub.uy/), Uruguay is among the top 5% countries in which the impacts of human development on biodiversity are expected to be the largest in the near future (Lee and Jetz, 2008).

2.2. Biodiversity features and datasets

The local conservation authorities identified 373 key biodiversity features for which reliable spatial distribution maps were also available (Suarez-Pirez and Soutullo, 2015) (Table 1 and Table S1 in Appendix A for a full list): i) 219 mammals, birds, amphibians, freshwater fish and plant species; ii) 92 threatened native ecosystems; iii) 6 ecosystem services; iv) 7 nationally recognized ecoregions; and v) 13 landscape units.

All species were threatened, either at the global or national level, or vulnerable to projected climate change in Uruguay (Soutullo et al., 2013, 2012b; Suarez-Pirez and Soutullo, 2015). Nationally threatened species have either a small population size or distribution in the country. Deductive models were used to model the distribution of all species (see e.g. Maiorano et al., 2006). Deductive distribution models use information about species-habitat associations based on literature reviews and expert knowledge (Brazeiro et al., 2012 and Supplementary Methods for more information). Deductive models were considered the most effective tool for modelling species distributions, as most priority species had well-understood relationships with accurately mapped habitat variables (Brazeiro et al., 2012).

Both ecoregions and landscape units delineate large homogeneous regions based on biophysical similarities. Yet, while ecoregions were delineated by integrating information on topography, soil types, land cover and species distributions (Brazeiro, 2015), landscape units were defined on the basis of the similarity of the landscape structure in terms of land cover and spatial patterns of the different landscape components (i.e., matrix, patches and corridors) (Evia and Gudynas, 2000). Threatened native ecosystems are those that currently cover <1% of Uruguay and are expected to further decline in size due to land-use change (Brazeiro et al., 2012). Threatened ecosystems represent smaller homogeneous units, which were mapped on the basis of land cover information derived from satellite imagery, soil types and topographic features and are mainly composed of native species.

For ecosystem services, we considered 6 provisioning and regulating services (Millennium Ecosystem Assessment 2005) that are relevant for rural activities, with benefits obtained in situ or in the vicinity of the ecosystems that produce them: (i) drinking water provision (continued access to water for consumption); (ii) genetic resources provision (maintenance of a high diversity of native organisms); (iii) climatic regulation (provision of conditions of temperature and humidity that are favourable for humans, cattle and most common local crops); (iv)

Fig. 1. Map of Uruguay showing existing protected areas, main roads, departments and water bodies.

water quality regulation (maintenance of the quality of water available for consumption); (v) natural hazard regulation (buffering of the negative impacts of floods, droughts and storms); (vi) disease and pest regulation (prevention of the spread of diseases or pests harmful to humans, cattle and crops). As for the other biodiversity features, the ecosystem services were mapped independently from this study (see Soutullo et al., 2012a for full details on how this was done), using the framework proposed by Maynard et al. (2010). Rather than formulating production functions to estimate the amount of ecosystem services produced in a certain area, the framework uses expert elicitation techniques to rank ecosystem types according to their relative contribution to the provision of each of the ecosystem services being considered (Maynard et al., 2010). The resulting maps show the relative contribution of each ecosystem type to the provision of that ecosystem service, with a value of 0 meaning no contribution and a value of 1 meaning maximum relative contribution (Soutullo et al., 2012a and Supplementary Methods for more information).

As the selected ecosystem services are relevant for rural activities across the whole country, and most of the benefits are provided in situ or in the vicinity of the ecosystems that produce them, we sought to identify areas of high ecosystem service provision within each of the seven main water catchments of the country. We did so as an attempt to prioritize not only areas that provide high levels of ecosystem services, but also to ensure that these areas are distributed in a manner that ensures availability of those services where there is human demand (Cimon-Morin et al., 2013). Hence, we treated each ecosystem service as a separate biodiversity feature for each main water catchment in the prioritization analysis (more details below).

Table 1
List of biodiversity features and corresponding weights assigned in Zonation.

Biodiversity features	# recorded in the country	# used in the analysis	Weight
Amphibians	48	21	0.457
Birds	455	55	0.457
Fishes	219	26	0.457
Mammals	117	30	0.457
Plants	2400	87	0.457
Ecoregions	7	7	14.286
Ecosystem services	–	42	2.381
Ecosystems	121	92	1.087
Landscape units	13	13	7.692
Total		373	

2.3. Spatial conservation prioritization

In order to identify the priority areas for the conservation of biodiversity and ecosystem services in Uruguay, we used the Zonation version 4.0 software (Di Minin et al., 2014; Moilanen et al., 2014). Compared to other conservation planning tools, Zonation produces a complementarity-based and balanced ranking of conservation priority over the entire landscape (Moilanen et al., 2011, 2005), rather than satisfying specific targets at minimum cost. The priority ranking is produced by iteratively removing the grid cell or planning unit that leads to the smallest aggregate loss of conservation value, while accounting for total and remaining distributions of features, weights given to features, and feature-specific connectivity. Detailed explanations about Zonation are provided in Di Minin et al. (2014), Lehtomäki and Moilanen (2013) and Moilanen et al. (2014).

Fig. S1 in Appendix A shows a flowchart of analysis and data inputs used in Zonation. All input data layers were used in their original resolution of 1 ha grid cells. The analysis was constrained to terrestrial areas that still retain a reasonable degree of naturalness, by masking out urban and other heavily modified areas (land use types A11 and B15) from Uruguay's land use map in 2011 (http://www.fao.org/3/a-i4372s.pdf). This was done in order to prioritize only in areas not heavily modified by human activities. While we considered spawning areas of freshwater fish in flood plains, we did not carry out an integrated terrestrial and freshwater conservation planning assessment.

The additive-benefit function cell removal rule for aggregation of conservation value was used in Zonation (see Moilanen et al., 2011). The additive-benefit function computes a maximum-utility type solution, where value is additive across biodiversity features, and where feature-specific representation is converted to value via concave power functions, which most commonly are parameterized according to the canonical species-area curve (Moilanen, 2007). Here, the exponent of the power function was set to $z = 0.25$ for all features. Compared to other cell removal rules available in Zonation (Di Minin et al., 2014; Moilanen et al., 2014), we chose the additive benefit function because no strict targets for protection were required (cf. target-based planning cell removal rule) and because a solution with the highest return on conservation investment (cells with high species richness) was required (cf. core-area removal rule) (Di Minin and Moilanen, 2012).

In Zonation, weights assigned to features influence the balance among features in the prioritization solution. Typically, weights have positive values, but can also be set to 0.0 in surrogacy analyses (Di Minin and Moilanen, 2014), or even have negative values, for example when multiple opportunity costs are included in the analysis (Di Minin et al., 2013; Moilanen et al., 2011). In the present analysis, we considered both positive (for biodiversity and ecosystem services) and negative (for potentially negative competing land uses, such as agriculture and commercial forestry, and land cost) weights. We did so to produce a spatial priority ranking that reduces the conflict between competing land uses (Moilanen et al., 2011). National conservation authorities considered all biodiversity features as equally important for conservation in Uruguay so that each feature was considered to have the same weight ($w_j = 1$) (MVOTMA, 2015). In order to avoid unequal aggregate weights based on different number of biodiversity features within each group (e.g. an aggregate weight of 219 for species and 92 for threatened ecosystems), potentially leading to the group with the largest number of biodiversity features having the greatest influence on analysis outcomes, we assigned the same aggregate weight ($W_j = 100$) to each group of biodiversity features and rescaled the weights of features within each group to sum up to the aggregate group weight (e.g. 7 ecoregions $*$ 14.286 $=$ 100) (Table 1) (see Lehtomäki and Moilanen, 2013 for more information how to rescale weights in Zonation). The construction of the Zonation algorithm is such that a sensible and efficient balance between features can be obtained even with the use of default weights ($w_j = 1$). An important factor in achieving such an outcome is the range-size renormalization, which adds emphasis to narrow range features.

In the analysis, we identified priority areas for conservation actions outside of the existing protected area network (Fig. 1), by using a hierarchical mask that identified the locations of 12 extant protected areas. This way it is guaranteed that the highest priorities are located in existing protected areas (Lehtomäki and Moilanen, 2013). In addition, we included in the analysis information on land ownership for all rural cadastral units in Uruguay. Specifically, we used 422,875 cadastral units as the units of selection, namely the planning units, in Zonation (Di Minin et al., 2014; Moilanen et al., 2014). This allowed us to produce ranked maps of landowners based on the importance of their properties in representing biodiversity and ecosystem services.

We developed four main scenarios: (i) business as usual, a scenario where only biodiversity and ecosystem services were included in the analysis; (ii) maintain agriculture and forestry, a scenario where we sought a balance between biodiversity and ecosystem services and land suitability for agriculture and commercial forestry; (iii) cost optimistic, a scenario where we additionally included land price in the analysis to reduce opportunity costs to local developers and other stakeholders in a balanced manner; and (iv) cost pessimistic, a scenario where we assigned higher weights to alternative land uses and land price to account for unsustainable development (Fig. S1 in Appendix A).

Under all scenarios, areas that were previously identified as potential reserves for fauna and flora by national policies (DGRNR, 2008) were given a positive weight of $w_j = 100$, equal to the aggregate weight of each group of biodiversity features. This was done so, as there are obvious opportunities for achieving conservation in these areas. In the maintain agriculture and forestry scenario, a map on soil suitability for agriculture and commercial forestry (DGRNR, 2008) was included as a negatively weighted ($w_j = -100$) feature in Zonation. These alternative land uses and biodiversity conservation have low compatibility. In the cost optimistic scenario, land price was included as a negatively weighted feature ($w_j = -50$), so that the aggregate weight with the other negatively weighted feature (soil suitability for agriculture and commercial forestry, $w_j = -50$) was $W_j = -100$. Land price was calculated as the mean value in US dollars paid for the properties sold in each judicial district between 2007 and 2010 (see Table S2 in Appendix A for total number of districts considered and range of land price values across districts) (DIEA, 2014). Areas where land price is high could make unfeasible the promotion of low intensity activities more compatible with conservation. We used land price as a proxy for the feasibility of implementing conservation actions. We did not use land price as a proxy for the economic cost of acquiring land for conservation, as this is not an option in Uruguay. In the cost pessimistic scenario, agriculture and commercial forestry and land price were assigned the highest weights ($w_j = -300$) in the analysis so that the aggregate weight for the negatively weighted features ($W_j = -600$) was equal to the aggregate weight of all positively weighted features ($W_j = 600$). In this case, we wanted to test how representation of biodiversity and ecosystem services would be affected by unsustainable development practices.

Zonation automatically produces a number of different output files for each run (Di Minin et al., 2014; Moilanen et al., 2014). Performance curves are automatically produced and exported for each feature during a Zonation analysis. These curves quantify the proportion of the original occurrences retained for each biodiversity feature at any fraction of the landscape chosen for conservation. Because there can be many of these curves, it is common to visualize curves as averages across groups of biodiversity features, as we did in this analysis. The priority rank map is the other main output of a Zonation analysis run. The priorities are derived from the order of iterative planning unit ranking. Thereby each cadastral unit in this map has a value between 0 and 1, with planning units with values close to 0 removed first because of their low conservation value (and possibly high cost, when cost is accounted for), and planning units with values close to 1.0 having high occurrence levels for many biodiversity features and ecosystem services. The priority rank map

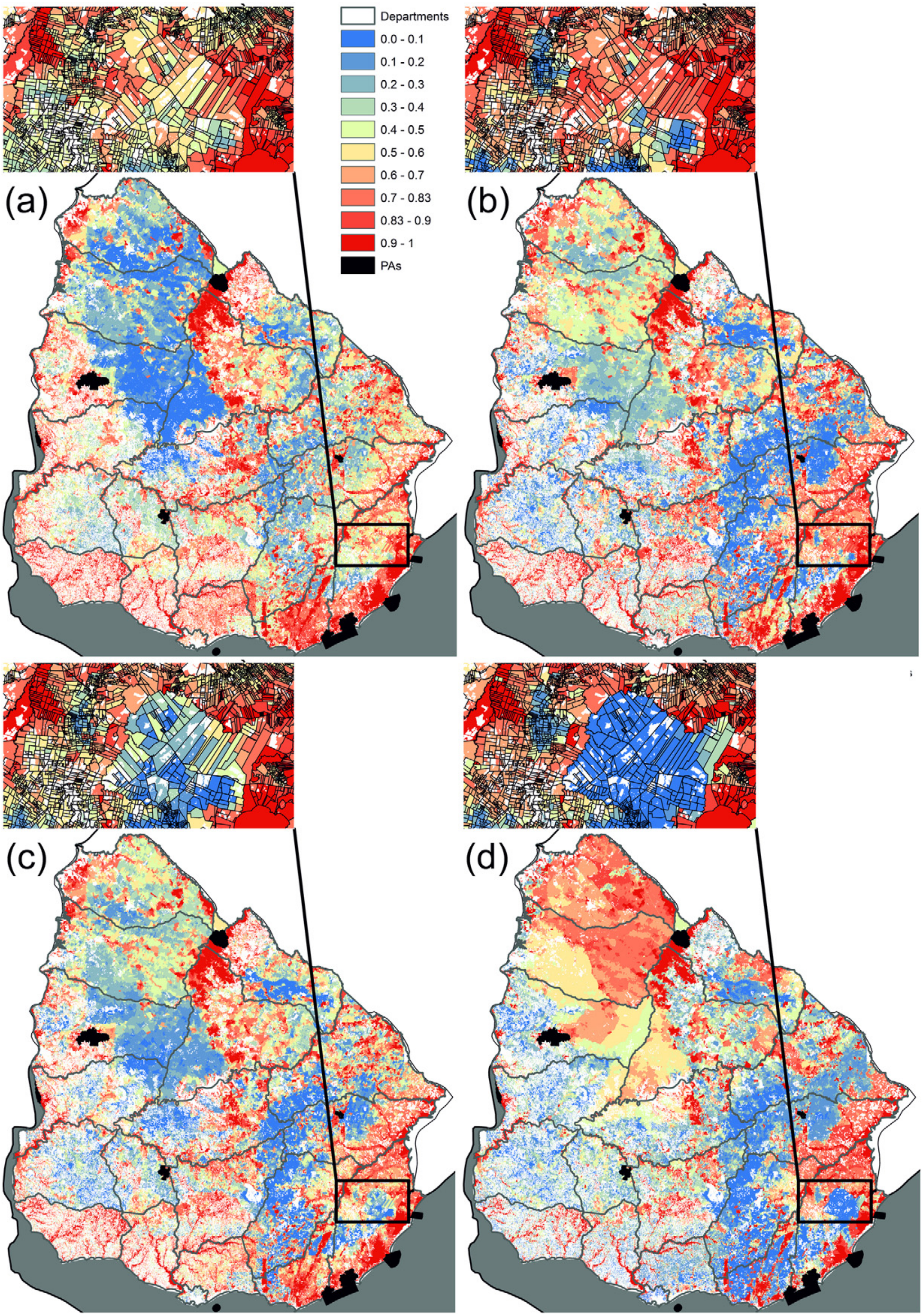

Fig. 2. Priority rank maps for the conservation of biodiversity and ecosystem services in Uruguay obtained by (a) including only biodiversity and ecosystem services; (b) biodiversity, ecosystem services and alternative land uses; (c) biodiversity, ecosystem services, alternative land uses and land cost; and (d) biodiversity, ecosystem services, and both alternative land uses and land cost with higher weights than in the previous scenario to consider unsustainable development practices. Areas in dark red are priorities for conservation actions. The insets show how the prioritization used cadastral units as planning units. PAs = protected areas. Departments represent local administrative boundaries within Uruguay. Areas in white are transformed.

corresponds directly with the performance curves. In this study, we show the representation of groups of biodiversity features summarized by the respective performance curves at the top 17% of the priority rank map, as this was the proposed target for terrestrial conservation in Uruguay (https://www.cbd.int/doc/meetings/cop/cop-11/official/cop-11-26-en.pdf).

A number of post processing analyses were also run. First, we identified the planning units that fell within the top 17% priority areas under each scenario and created a 'consensus' priority rank of landowners across all scenarios. We did this to assess the representation of biodiversity features and ecosystem services in areas that were identified as priorities across all scenarios and that are therefore priorities for conservation actions. Second, we calculated the total land cost for each proposed land target under each scenario, including the 'consensus' map, by (i) intersecting the priority rank map with the land price maps (mean, as well as minimum and maximum value maps to account for uncertainty) and (ii) multiplying the values across all identified planning units in ArcMap (ESRI 2011). Finally, we ran *t*-tests in R v. 3.1.1 (R Core Development Team, 2016) to determine whether the means of two groups of biodiversity features from 2 separate scenarios were equal or not to each other.

3. Results

The southern western, eastern and central northern parts of Uruguay were identified as priority areas for the conservation of biodiversity and ecosystem services under all scenarios (Fig. 2). However, there was a shift in priorities away from some of these areas to the north western part of Uruguay when alternative land uses (Fig. S2b, Appendix A) and land price (Fig. S2c and d, Appendix A) were included in the analysis. This was particularly so under the cost pessimistic scenario (Fig. S2d, Appendix A). The shift in priority areas is also apparent in the priority ranking of landowners in insets of Fig. 2. At the 17% land target for conservation, the overlap between the business as usual scenario and the scenario where agriculture and commercial forestry were included in the analysis was 75%, the overlap between the business as usual scenario and the cost optimistic scenario was 85%, and the overlap between the business as usual scenario and the cost pessimistic scenario was 73%. Overall, the overlapping priority areas across all scenarios corresponded to 12.6% of Uruguay (Table S3 and Fig. S3, Appendix A).

At the 17% land target, the representation levels for each group of biodiversity features targeted for conservation were, on average, higher under the business as usual scenario (with the exclusion of landscape units) than they were under the other scenarios (Table 2; Fig. 3). Under the cost pessimistic scenario, the representation of all groups of biodiversity features was the lowest (Table 2; Fig. 3). The maintain agriculture and forestry ($t = 1.7$, $df = 178.679$, p-value $= 0.02$) and the cost pessimistic ($t = 1.8$, $df = 178.528$, p-value $= 0.01$) scenarios provided the lowest return on investment for threatened ecosystems, as the representation was significantly lower under these scenarios than it was under the business as usual scenario (Table 2; Fig. S4, Appendix A). The total opportunity cost was the highest under the business as usual scenario (6.84 billion USD, minimum of 2.74 and maximum of 14.90) and the lowest under the cost pessimistic scenario (5.54 billion USD, minimum of 2.37; maximum: 11.42) (Table S3 and Fig. S5, Appendix A). The conflict with agriculture and commercial forestry was also the highest under the business as usual scenario and the lowest under the cost pessimistic scenario (Table 2). Under the cost pessimistic scenario, the number of identified landowners was the smallest, but the average size of the property was the biggest (Table S3, Appendix A).

In order to meet the highest representation levels for each group of biodiversity features, as achieved across all scenarios (Table 2), a 2% increase in land target would be needed under the business as usual scenario (Fig. 3A), a 7% increase would be needed under the maintain agriculture and forestry scenario (Fig. 3B); a 3% increase would be needed under the cost optimistic scenario (Fig. 3C); and a staggering 41% increase would be needed under the cost pessimistic scenario (Fig. 3D). Under the expanded targets for conservation, the total opportunity cost was the lowest under the cost optimistic scenario (6.96 billion USD at the 20% land target, minimum of 2.99 and maximum of 14.25) and the highest under the cost pessimistic scenario (10.11 billion USD at the 58% land target, minimum of 4.70 and maximum of 18.70). Remarkably, the total opportunity cost at the 20% land target for conservation, under the cost optimistic scenario, was the same as the total opportunity cost at the 17% land target, under the business as usual scenario (Figs. S5 and S6, Appendix A). Under the cost pessimistic scenario, instead, the opportunity cost was 3.3 billion USD higher than the opportunity cost at the 17% land target under the business as usual scenario

Table 2
Average representation for each group of biodiversity features at the 17% land target for conservation under each scenario.

	No cost			Agriculture & forestry			Cost Optimistic			Cost Pessimistic		
	Min	Av	Max	Min	Av	Max	Min	Av	Max	Min	Av	Max
		17%			17%			17%			17%	
Species	0.111	0.613	1.000	0.091	0.597	1.000	0.118	0.613	1.000	0.091	0.589	1.000
Ecoregions	0.067	0.221	0.389	0.085	0.212	0.381	0.080	0.213	0.353	0.100	0.199	0.377
Ecosystem services	0.098	0.384	0.710	0.105	0.358	0.763	0.105	0.350	0.676	0.143	0.322	0.649
Landscape units	0.055	0.367	0.992	0.063	0.374	0.959	0.062	0.374	0.959	0.082	0.357	0.939
Ecosystems	0.078	0.654	1.000	0.026	0.571	1.000	0.053	0.629	1.000	0.025	0.566	1.000
Agric. & Forestry	–	0.163	–	–	0.073	–	–	0.113	–	–	0.069	–
		19%			24%			20%			58%	
Species	0.116	0.622	1.000	0.137	0.680	1.000	0.136	0.642	1.000	0.095	0.765	1.000
Ecoregions	0.070	0.230	0.403	0.125	0.289	0.509	0.093	0.242	0.395	0.197	0.506	0.919
Ecosystem services	0.103	0.397	0.728	0.160	0.440	0.811	0.123	0.384	0.709	0.431	0.575	0.863
Landscape units	0.057	0.374	0.993	0.098	0.463	0.985	0.073	0.404	0.976	0.322	0.671	0.951
Ecosystems	0.086	0.665	1.000	0.036	0.654	1.000	0.063	0.660	1.000	0.034	0.654	1.000
Agric. & Forestry	–	0.174	–	–	0.109	–	–	0.131	–	–	0.104	–

Note: light grey cells highlight the highest representation for each group of biodiversity features across all scenarios, while dark grey cells highlight the group of biodiversity features particularly affected under that scenario, requiring larger land conservation target.

(Fig. S6, Appendix A). Under the cost optimistic scenario, the number of identified landowners was the smallest, while, under the cost pessimistic scenario, the average and maximum property size were the biggest (Table S3, Appendix A).

The total opportunity cost to 27,530 landowners identified within the 'consensus' overlapping areas equalled to 4.8 billion USD (minimum of 1.96 and maximum of 10.27) (Table S3, Appendix A). Within these areas, 74–89% targets of the highest representation for all groups of biodiversity features could be potentially achieved (Table S4, Appendix A).

4. Discussion

In this study, we identified priority areas and landowners for conservation actions, while seeking to reduce the conflicts with agricultural and forestry production and opportunity costs of conservation to local landowners in Uruguay. Our framework integrates data on biodiversity features, ecosystem services, maps on agriculture and commercial forestry potential and information on land price to produce ranked maps of landowners to engage in conservation actions. We found that a small increase in the proposed conservation target at no additional cost to local landowners would maximize the representation of biodiversity and ecosystem services. Overall, our results benefitted from an almost unconstrained initial state, in which < 1% of Uruguay is currently protected. Threatened ecosystems should be targeted for immediate conservation actions as they are likely to be lost because of agricultural and commercial forestry expansion. We identified ~28,000 landowners across all scenarios that local conservation authorities should engage using targeted conservation actions.

Previous studies showed how including economic costs in conservation planning can help achieve higher return on investment than when planning ignores costs (see e.g. Bode et al., 2008; Carwardine et al., 2008). While alternative land uses have been included before in

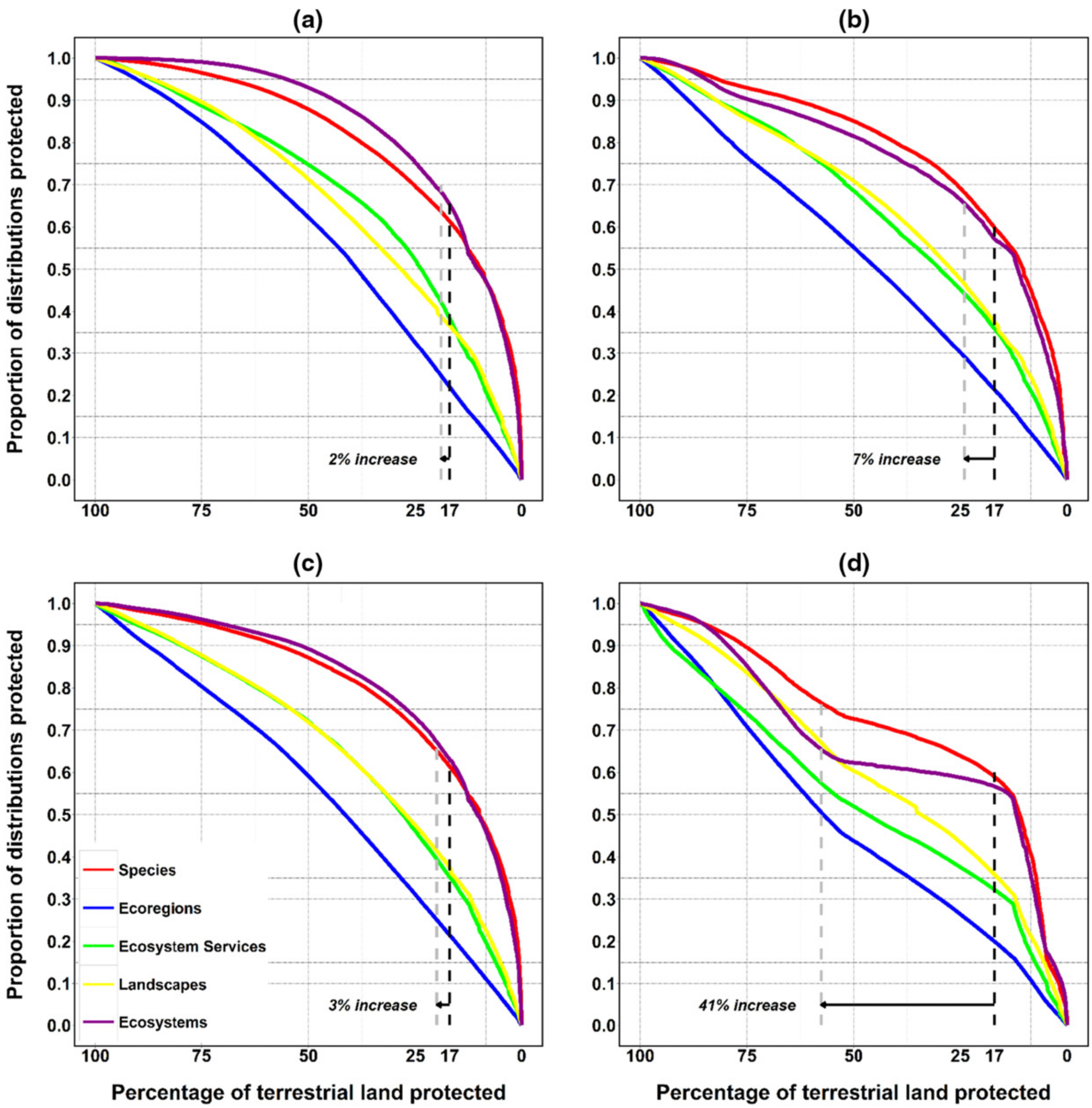

Fig. 3. Performance curves quantifying the average proportion of the original occurrences of each group of biodiversity features included in the analysis, represented at each fraction of the terrestrial land protected. In scenario (a) only biodiversity and ecosystem services were included in the analysis; in scenario (b) biodiversity, ecosystem services and alternative land uses were included; in scenario (c) biodiversity, ecosystem services, alternative land uses and land cost were included; and in scenario (d) biodiversity, ecosystem services, and both alternative land uses and land cost with higher weights than in the previous scenario to consider unsustainable development practices. The dashed vertical line in black represents the 17% target for terrestrial land conservation, as this is the proposed conservation target under Aichi target 11. The dashed vertical line in grey represents the percentage of terrestrial land required to meet the best representation levels for each targeted group of biodiversity features, as achieved at the 17% across all scenarios.

Zonation to reduce policy conflicts (Di Minin et al., 2016, 2013; Dobrovolski et al., 2014; Faleiro and Loyola, 2013; Moilanen et al., 2011; Montesino Pouzols et al., 2014; Nin et al., 2016), our framework can be used to rank landowners to potentially engage in conservation actions. Such ranked maps of landowners can be more directly linked to land use planning in order to promote stakeholders' engagement and enhance the implementation of conservation actions. Furthermore, the ranked maps can be used by national and local planners to prevent harmful development in top ranked properties for the conservation of biodiversity and ecosystem services. Our outputs, in fact, provide information on what biodiversity features are potentially present in each cadastral unit and the rank of that cadastral unit. Building on Moilanen et al. (2011), our framework also accounts for local demand for ecosystem services at a national scale.

We found that threatened ecosystems are likely to be lost in areas where agriculture and commercial forestry potential are high. Many of these ecosystems are temperate grasslands, which are the most threatened biome at the global level with <0.6% of its extent protected (Noss, 2013). Hence, immediate action is needed to engage landowners where these ecosystems occur and provide them with targeted incentives for conservation. Ecosystem services, which provide important benefits to humans, are also likely to be affected in these areas. As such, payments for ecosystem services to offset opportunity costs of conservation might be an important incentive for conservation in these areas. While it remains difficult to fully assess the economic value of some of the benefits generated by the ecosystems we considered (Nelson et al., 2009), the loss of the services they provide could potentially trigger societal costs that could outweigh the opportunity costs to local stakeholders. Future studies should fully assess the economic benefits generated by ecosystem services in the country in order to implement a full cost-benefit analysis (Cimon-Morin et al., 2013).

As Aichi targets are now integrated into almost all UN Sustainable Development Goals, decision makers are required to integrate ecosystem and biodiversity conservation into national planning to support sustainable development goals. Our framework can be used by decision makers in other developing countries for this purpose. An important limitation in other developing countries might be the lack of reliable data on the distribution of biodiversity features, ecosystem services, alternative land uses and cost. However, new datasets on biodiversity and ecosystem services are becoming increasingly available at the resolution needed to inform real-world conservation decision-making (see e.g. Mulligan et al., 2010; Jetz et al., 2012; Robertson et al., 2014). More funding is also being made available to biodiversity-rich, data-poor, developing countries to develop comprehensive, up-to-date information about the distribution of biodiversity and ecosystem services (Brooks et al., 2014). Our results suggest that investing in cost-effective methods to model distributions of biodiversity and ecosystem services and including this information in conservation planning assessments like the one we developed here might help prevent unsustainable development that can carry higher costs to the national economy and human well-being.

While our assessment was developed in collaboration with national conservation authorities and used information that was deemed relevant for identifying priorities for conservation in Uruguay (MVOTMA, 2015), it is important to acknowledge aspects of the study that could be improved. First, ecosystem services were mapped using an expert elicitation technique, which can be used to make relatively fast assessments under time constraints, but can suffer of high levels of subjectivity and does not provide quantitative estimates of ecosystem services (Martínez-Harms and Balvanera, 2012). Second, while our framework only included provisioning and regulating services, it can also be used to include a range of other ecosystem services (Chan et al., 2006; Hausmann et al., 2015; Naidoo et al., 2008). Third, expert-based approaches to map species distributions are cost and time effective (Johnson and Gillingham, 2005; Maynard et al., 2010), but suffer from a number of limitations (Graham and Hijmans, 2006) that might require accounting for uncertainty (Beale and Lennon, 2012; Guisan et al., 2013). Fourth, factors such as landowners willingness to sell (Knight et al., 2011) or political willingness to act (Faleiro and Loyola, 2013), beyond land price, could have been used in the analysis. However, this information is easier to collect at the regional level and/or requires longer time to collect at the national level while decisions need to be taken quickly.

In conclusion, minimizing conflict with agriculture and commercial forestry and reducing opportunity costs to local landowners is possible, but challenging even in a country where current protected area coverage is so small. Losing threatened ecosystems and ecosystem services will reduce the benefits society accrues at large. As such, a small addition to the proposed conservation target at no additional cost to local landowners will provide long-term benefits to human well-being and the Uruguayan economy. Priority areas and landowners where threatened ecosystem and ecosystem services are found should be targeted for immediate conservation actions by national conservation authorities. Our cadastre-based country-wide conservation planning assessment, including biological and socio-economic data at the scale needed to inform conservation decision-making, can be adapted to include other types of data and repeated elsewhere to inform global and national conservation policy and the UN Sustainable Development Goals.

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Appendix A. Supplementary methods and results

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.biocon.2016.11.037.

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